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Consequential life cycle assessment of urban source-separating sanitation systems complementing centralized wastewater treatment in Lund, Sweden

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A R T I C L E I N F O

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A B S T R A C T

This study examined various source-separating sanitation systems to evaluate their environmental performance, providing decision-makers with insights for selecting an appropriate system for a newly developed neighborhood in Sweden. A full consequential LCA was conducted to account for resource recovery and substitution. The local wastewater treatment plant WWTP was modeled as a reference. Secondly, a urine recycling system was introduced to treat 75 % of the collected urine, with the remainder piped to the WWTP. Thirdly, a black and greywater (BW&GW) treatment system handling all generated wastewater was examined. Finally, a hybrid source-separating system combining urine, black, and greywater was investigated. The results indicated that the four scenarios exhibited global warming potentials (GWP) of 78, 62, 32, and 24 kg CO2-eq per PE/ y. Recycling urine as fertilizer led to a 20 % reduction in the GWP of the reference. It also reduced other impact categories, with a 55 %, 65 %, and 45 % reduction in eutrophication, ozone depletion, and acidification, respectively. The BW&GW system achieved a 60 % reduction over the reference GWP, mainly due to fertilizer, biogas, and cleanwater recovery. Integrating urine, black, and greywater recycling in the final scenario achieved a 25 % reduction compared to the BW&GW scenario, primarily due to lowering of the ammonia stripping GWP and the additional fertilizer recovery. Based on sensitivity analyses, switching citric acid for sulfuric acid reduced the GWP of the urine stabilization unit process by 101 %, from 15.47 to -0.14 kg CO2-eq per PE/ y. Ultimately, the findings suggest that the fully decentralized source-separating sanitation system incorporating urine, blackwater, and greywater recycling, particularly when combined with 70 % energy recovery at the urine concentrator, is most favorable.

1. Introduction

Domestic wastewater is loaded with resources that can be recovered in different forms (e.g., biogas, fertilizer, and clean water) instead of being discharged into the environment, causing adverse environmental impacts (Malila et al., 2019). These pressures, such as eutrophication, climate change, acidification, and ozone depletion, are evident examples of the growing future uncertainties that threaten the well-being of our ecosystems (Rockstrom et al., 2023). To alleviate these threats and move forward to achieve sustainable development goals (SDGs) while keeping the planetary boundaries within their thresholds, today's wastewater management systems need to incorporate circularity and close resource loops (Larsen and Binz, 2021; Trimmer Jt Cusick, 2017). Various experts have examined and regarded source-separating sanitation systems (i.e., the separate collection and processing of wastewater fractions) as a potential alternative to conventional wastewater treatment for maximizing resource recovery in the sanitation sector (McConville. et al., 2017).

Several source separation methods and systems have been developed worldwide for the separate collection and treatment of different wastewater fractions (Aliahmad et al., 2022; Harder et al., 2019; Larsen et al., 2021). These systems were found to not only foster circularity and promote resource recovery (Fam and Mitchell, 2013) but also to have the potential to reduce nutrient and micropollutant emissions from wastewater treatment plants (WWTPs) (Badeti et al., 2021) and lower energy and financial costs (Igos et al., 2017). Some concrete models of

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the source-separating sanitation systems are urine and blackwater recycling (Sniatala et al., 2023). Blackwater (containing feces, urine, flush water, and toilet paper) accounts for only 15 % of the total domestic wastewater volume yet contains approximately 90 % of the nitrogen and 80 % of the phosphorus (Saliu and Oladoja, 2021). Urine, on the other hand, is even more concentrated, at about 1 % of domestic wastewater volume and containing approximately 80 % of the nitrogen and half of the phosphorus and potassium (Jönsson, 2005; Vinnerås et al., 2006). The separate collection and recycling of blackwater and/or urine thus offers the prospect of increasing nutrient recovery, meeting expected phosphorus and nitrogen recovery targets in Sweden, while at the same time reducing the carbon footprint of sanitation management in support of existing national Swedish environmental goals related to climate change (Lehtoranta et al., 2022a; McConville et al., 2017). Additionally, nutrient recovery from domestic wastewater can potentially reduce reliance on agricultural mineral fertilizers (Lehtoranta et al., 2022a; Saliu and Oladoja, 2021). Contemporary intensive farming methods rely heavily on these fertilizers, which are rich sources of phosphorus and nitrogen (Sniatala et al., 2023). Their price depends upon the cost of phosphate extraction and the natural gas used in the fixation of nitrogen in the Haber–Bosch process (Kok et al., 2018; Langergraber and Muellegger, 2005). Therefore, any volatility, such as geopolitical tensions, can create dramatic price swings. Since mineral phosphorus is also relatively scarce and the reserves of fossil fuels will soon run out, these nutrients are likely to become too expensive to capture (Cordell et al., 2009), posing a threat to the prosperity of countries susceptible to economic shock and those which rely on fertilizer imports.

While these source-separating sanitation systems have been explored from a technical perspective and optimized to maximize resource recovery (Kjerstadius et al., 2015; Mehaidli et al., 2024; Simha et al., 2018; Tarpeh et al., 2017; Udert et al., 2003), and from a socio-technical perspective to identify diffusion barriers (Abeysuriya et al., 2013; Aliahmad et al., 2023; McConville et al., 2023; Simha et al., 2021), less emphasis has been placed on exploring their comparative environmental profiles (Aliahmad et al., 2022; Mathilde Besson and Tiruta-Barna, 2021). Considering that these systems aim to improve wastewater sustainability and mitigate emerging uncertainties, their environmental profiles and foreseeable consequences must be thoroughly examined to decide whether they are sustainable alternatives.

The life cycle assessment (LCA) methodology has been employed to study and evaluate the environmental profiles of conventional wastewater treatment and source-separating sanitation systems. In turn, this has contributed to a better understanding of the environmental performance of these systems throughout their life cycle, providing insights for decision-makers involved in the strategic planning of urban infrastructure (Heimersson et al., 2019). Some of these LCA studies have focused on conventional WWTPs (Corominas et al., 2020; Raghuvanshi et al., 2017), the environmental implications of the end products (Lam et al., 2022), and the associated environmental trade-offs (Pausta et al., 2024). Some have extended their analysis beyond centralized WWTP and compared it to decentralized systems (Risch et al., 2021) or examined different spatial scenarios, including developing countries (Gallego-Schmid and Tarpani, 2019) and small communities (Garfí et al., 2017). On the other hand, fewer studies have focused on comparing source separation systems, such as blackwater systems, with conventional systems (Kjerstadius et al., 2017; Lima et al., 2023; Remy, 2010; Thibodeau et al., 2014). There has also been partial investigation into other source separation systems, including urine recycling (Ishii and Boyer, 2015), fertilizer production (Hilton et al., 2021; Martin et al., 2023), and life cycle costing (Landry and Boyer, 2016). Recent LCAs have demonstrated that source separation systems, such as urine recycling and blackwater, outperform conventional WWTPs regarding environmental impact (Besson et al., 2021). This is often attributed to the additional resources these systems recover as well as a reduction in greenhouse gas emissions such as nitrous oxide N_2O (Benetto et al., 2009; Lundin et al., 2000). However, there is a noticeable gap in large-scale comparative studies on these systems, as most studies have focused on smaller or semi-large scales (Besson et al., 2021; Spångberg et al., 2014). Existing studies, though informative, have limitations in their comparative scope; for example, none have investigated the potential benefits of a hybrid/integrated source-separating system of urine and blackwater. Ammonia stripping, for instance, was reported as a primary source of climate impact in the blackwater system (Lima et al., 2023), highlighting the need to explore whether incorporating urine recycling would mitigate this impact. Furthermore, to the best of our knowledge, most of the LCA studies reviewed are attributional, meaning they used average data in their analysis. This underscores the need for further comparative consequential LCA studies on a larger scale.

Therefore, the primary aim of this study is to address existing research gaps by performing a full consequential life cycle assessment (CLCA) on different source separation scenarios, including blackwater, urine, and a hybrid scenario of both in a large-scale newly built neighborhood of 10,000 person-equivalent in southern Sweden. Herein, the study is structured to address the following research questions: 1. *What are the foreseeable environmental impacts of conventional WWTPs compared to source separation systems throughout their life cycles? 2. What environmental hotspots are associated with each source separation scenario, and how can these be mitigated? What sets this LCA apart is the utilization of the consequential LCA approach, utilizing marginal data to model the environmental gains of substituting conventional resources with recovered products such as fertilizer, biogas, and water, details of which are further elaborated within the study. The CLCA approach aligns with the LCA's overarching goal, which is to assist decision-makers in selecting an appropriate source separation system for the newly constructed Brunnshög neighborhood in the city of Lund, located in the south of Sweden by illustrating the environmental consequences associated with these systems in comparison to a centralized WWTP. Using the CLCA methodology enables the inclusion of both direct and indirect impacts, allowing us to capture the foreseeable environmental consequences of adopting a specific sanitation system.*

2. Material and methods

2.1. Case study

The study is conducted in the city of Lund, in southern Sweden. The specific location is Brunnshög, a newly developed, under-construction neighborhood planned to house 40,000 people by 2050 (Brunnshög, Lund Kommun, 2024). The wastewater in Lund is currently being treated in the local Källby wastewater treatment plant (WWTP). However, this treatment plant is planned to shut down in the near future, and wastewater will be treated in the Sjölunda WWTP. However, Sjölunda WWTP in Malmö city has now reached a point where it would need extensive renovation to receive more wastewater. Proposing source separation sanitation systems to handle the wastewater generated in Brunnshög would potentially bring environmental benefits to the centralized WWTP and contribute to the ecological profile of the neighborhood. The proposed demo site in Brunnshög is assumed to cover 4000 apartments, hosting a total of 10,000 person-equivalent (PE).

2.1.1. Description of scenarios evaluated

In this LCA, we examined four distinct types of urban sanitation systems. The comparison revolves around centralized sewage conveyance and treatment with alternative scenarios of decentralized and semi-centralized sewage treatment that also involve different extents of source-separation of sewage. In the first scenario, a conventional WWTP serves as a baseline for comparison with other scenarios. A schematic diagram illustrating the WWTP's operation can be found in Fig. 1. In this diagram, we depict the WWTP in operation in Helsingborg City, which was selected due to its relevance and capacity size, which is similar to Lund. We have modeled the Helsingborg and the existing Sjölunda

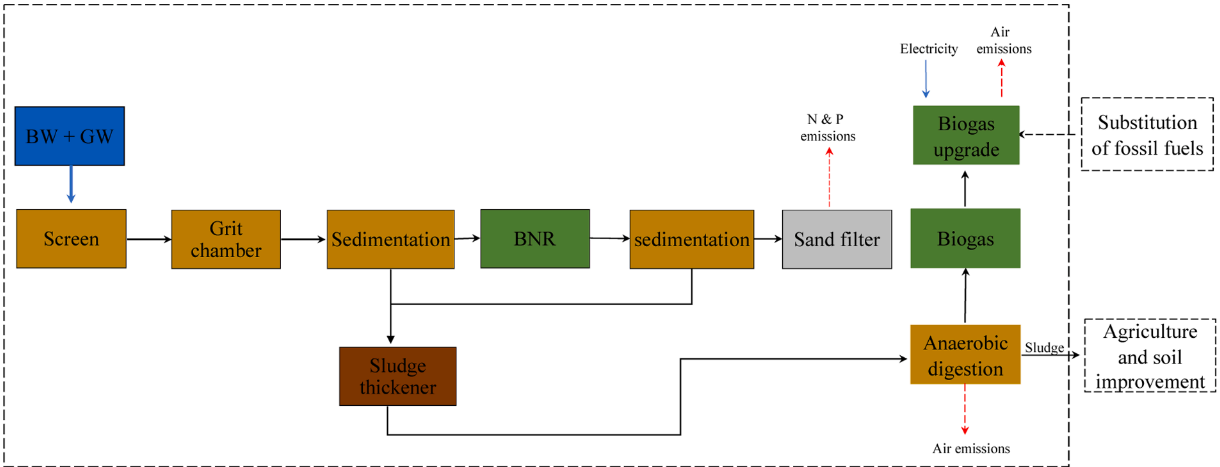

Fig. 1. Schematic diagram of Helsingborg wastewater treatment facility.

WWTPs to compare their environmental performance before proceeding with the former.

For the WWTP, blackwater and greywater (BW& GW) are mixed and collected inside the buildings in one pipe and transported through the sewer network to the facility, as shown in Fig. A.1. The influent undergoes several treatment steps, reducing and removing the biochemical oxygen demand (BOD), chemical oxygen demand (COD), and nutrients (Nitrogen and Phosphorus). Biogas is produced and upgraded to substitute diesel in buses; sludge is also produced, half of which is used in agriculture fertilizer and the other half as soil conditioner.

The second scenario incorporates the concept of urine recycling, i.e., the separate collection and treatment of urine from other wastewater fractions using a urine-diversion toilet (UDT). It is assumed that 75 % of urine is collected (the efficiency of the UDT) (Gundlach et al., 2021). To ensure comparability between the different scenarios, 25 % of the uncollected urine and the rest of the wastewater (grey and brown water) are accounted for in this scenario and assumed to be sent to the local WWTP in a second pipe. We have adjusted the WWTP to account for nitrogen and phosphorus reduction. This scenario is illustrated in Fig. A.2 for visual representation and further details. As part of this setup, urine undergoes pretreatment in the building basement in order to stabilize it, i.e., keep nitrogen as urea by inhibiting its hydrolysis into ammonia by reducing pH to $\leq$ 3.0 with the addition of an organic/inorganic acid (Simha et al., 2023). After urine is stabilized, it is concentrated to remove water and achieve a 95 % reduction in mass. The water is assumed to be recovered using a heat exchanger that also recovers 60–80 % of the heat used in concentrating the urine (Simha et al., 2020). The 60–80 % energy recovery range was selected based on the feasibility of achieving this in residential settings using well-established technologies like air-to-air heat exchangers and heat pumps. Literature on wastewater heat recovery, including (Wehbi et al., 2023), suggests a typical heat recovery of 50–60 % in residential applications. Additionally, (Larsen et al., 2021) report that the energy required for treating urine by distillation is 110 Wh$\cdot L-1$, compared to 710 Wh$\cdot L-1$ for water evaporation without energy recovery. Thus, the assumption of 60–80 % energy recovery is reasonable and reflects a range achievable with existing systems. The concentrated urine is subsequently transported to a factory, where it is fully dehydrated by vacuum drying and pelletized to produce solid fertilizer that can replace mineral fertilizers (as shown in the complete schematic diagram in Fig. 2).

In the third scenario, 100 % black and greywater are recycled. This system mimics the existing pilot system *H+* in Helsingborg; for a detailed understanding of the system, readers are directed to (Kjerstadius et al., 2015). This configuration's environmental profile has been studied previously (Lima et al., 2023; Remy, 2010), though we have altered it to accommodate new population equivalents (PE) and wastewater characteristics and have chosen not to include food waste recycling, a component that was considered in their studies (see Fig. 3). An advantage of this design over the previous two is that it features a fully decentralized sanitation system, eliminating the need to pipe wastewater to a central wastewater treatment plant. This scenario is

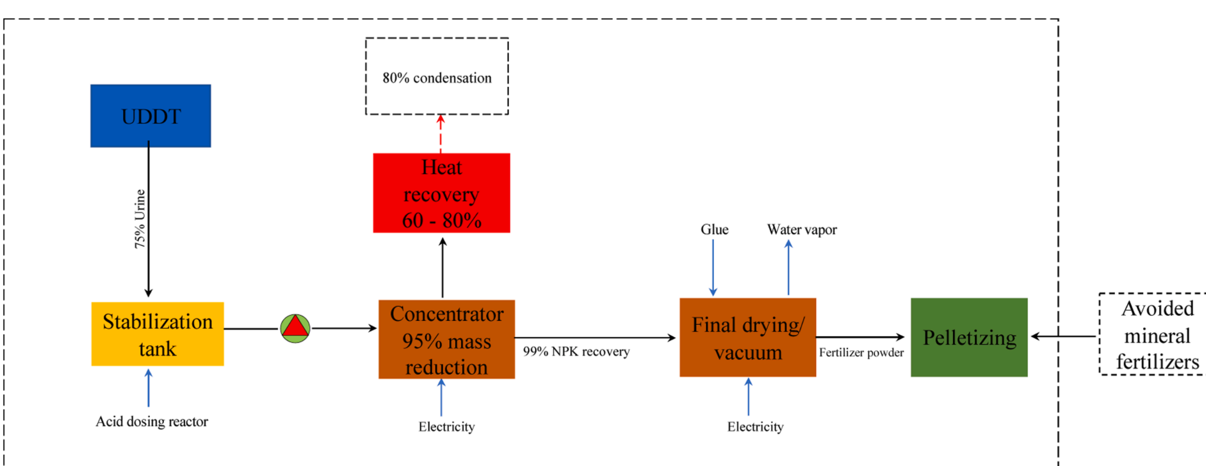

Fig. 2. Schematic diagram of the urine recycling system.

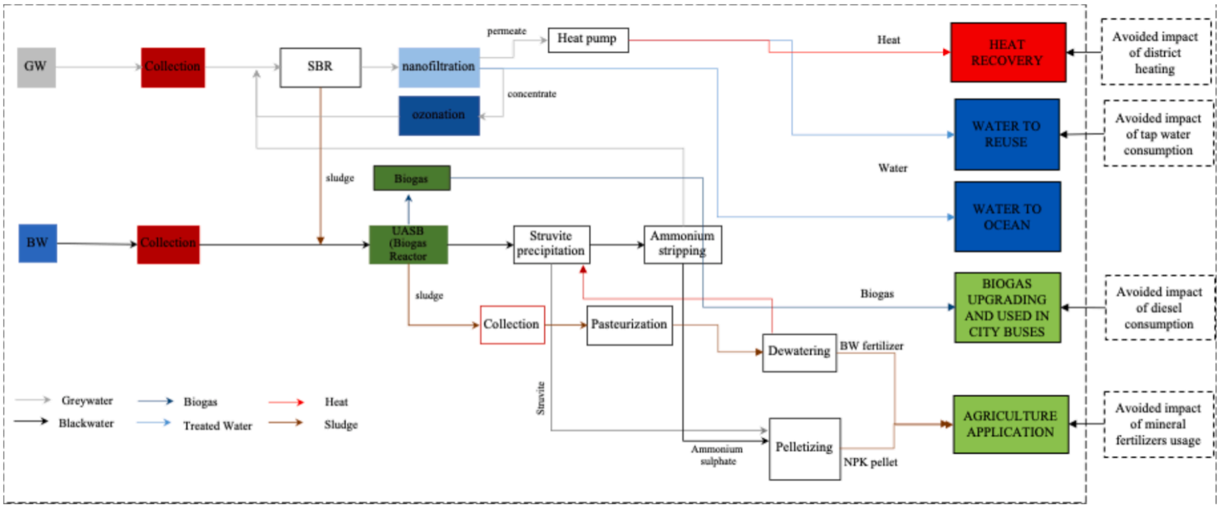

Fig. 3. Schematic diagram of the black and greywater recycling system.

illustrated graphically in Fig. A.3. The blackwater undergoes a series of treatments, including up-flow anaerobic sludge blanket digestion (UASB), which produces biogas and sludge. The UASB effluent is then further processed by struvite precipitation and ammonium stripping to recover phosphorus and nitrogen in the form of struvite and ammonium sulfate, which can be made into NPK fertilizer. In addition, after the fertilizer's recovery, the left digestate is collected and transported to be applied in farmland. The sludge from the UASB is subsequently pasteurized and then dewatered into biofertilizers, which, together with the NPK fertilizer, can replace mineral fertilizers in agriculture. The biogas is upgraded to a quality suitable for use in city buses. Concurrently, greywater is treated in a sequencing batch reactor (SBR), followed by a series of disinfection processes of nanofiltration and ozonation. The ozonation effluent is recirculated back to the SBR while the permeate passes through a heat pump, where heat and water are recovered and reused. The sludge from the SBR process joins the blackwater stream before the UASB. Despite the high quality of the reclaimed water, the regulatory restraints in Sweden and the absence of explicit permits necessitate the discharge of 20 % of the treated black and greywater into the ocean. The remaining 80 % is utilized for irrigation purposes (Lima et al., 2023).

The fourth scenario, illustrated in Fig. A.4, integrates the previously discussed urine recycling and blackwater systems. Similar to the previous scenario, this scenario also provides the advantage of treatment being fully decentralized, thereby avoiding the need for piping uncollected wastewater to a central WWTP. According to (Lima et al., 2023), ammonia stripping was a primary source of climate impact in the blackwater system in Helsingborg. In this final scenario, we examine whether the collection and treatment of urine, which contains the majority of nitrogen, helps to improve the blackwater system in terms of climate impact. Practically, as shown in the illustration, there are three separate pipes exiting the building in this scenario: one for the diverted urine, which is treated according to the method outlined in the second scenario; one for the uncollected urine, as well as the remaining blackwater; and one for greywater, which will be treated following the same procedures as the third scenario.

2.2. Life cycle assessment LCA

The International Standard 14,040 established a standardized methodology for life cycle assessment (LCA), which analyzes and quantifies the potential environmental impact of a product, from extraction to disposal ("ISO 14040," 2006). This methodology is not only a theoretical construct but is a practical tool that guides one through four main phases: defining a goal and scope, determining a life cycle inventory, assessing a life cycle impact assessment, and interpreting the results. Phases are not isolated but are interconnected, with each building upon the previous. Through this iterative process, alternatives under investigation are selected, and environmental hotspots are identified.

In general, life cycle assessment (LCA) involves two methodological alternatives: attributional and consequential. Choosing between attributional and consequential modeling is essential to the results of an LCA study because both approaches address a specific question, and an adequate choice makes the analysis and results more consistent with the decision context (Tillman, 2010; Weidema, 2003). An Attributional Life Cycle Assessment (ALCA) identifies a product's direct environmental impact (emissions). ALCA utilizes average data that is representative of the actual physical flow of products (Finnveden et al., 2009). Alternatively, the Consequential Life Cycle Assessment (CLCA) incorporates indirect emissions into the analysis, taking into account the more systematic changes caused by the product's decision (i.e., use and operation) (Curran, 2007; Ekvall, 2020). As part of a CLCA, upstream and downstream changes in supply chains are analyzed, as are market-driven factors such as changes in production, consumption, and substitution (Ekvall T, 2004; Sandén and Karlström, 2007). A CLCA utilizes marginal data to determine the additional environmental impact associated with the production and introduction of an additional unit of a product (Weidema BP, 1999; Zamagni et al., 2012).

Regarding multifunctionality—multiple outputs from a single process—the two approaches to quantifying emissions differ significantly. A specific allocation method is used in ALCA to partition the impacts based on set criteria among the outputs (Azapagic 1999), whereas system expansion avoids allocation in CLCA (Ekvall and Andrae, 2005; Wernet et al., 2016). Two approaches to system expansion may be utilized: one approach involves expanding the system boundaries to include a new function or product, harmonizing the scope of the systems being compared (Earles and Halog, 2011). An alternative to this method, the "avoided burden" method, subtracts the environmental burdens resulting from an alternative method of providing the secondary function from the overall system (Ekvall, 2020; Ekvall et al., 2016). The latter is what we used in this study as it was deemed appropriate in the context of wastewater treatment (Tillman, 2010).

2.2.1. Goal and scope definition

The primary goal of this LCA study is to evaluate and compare different source-separating sanitation systems for a newly developed neighborhood in southern Sweden against the local centralized WWTP.

The study aims to identify environmental hot spots, which will be essential for optimizing proposals and recommendations for implementation. The study is focused on a specific case area in Sweden with its current reference system where biogas is produced and upgraded to substitute diesel in buses, and sludge is also produced and used in agriculture fertilizer and soil conditioners. Therefore, this study is not meant to compare what is best going forward by either the Water Resource Recovery Facility (WRRF) or source separation but instead compare source separation to the existing local WWTP. Sanitation systems are generally designed for managing and treating incoming wastewater. Accordingly, this LCA's functional unit (FU) is the management of domestically generated wastewater per person equivalent (PE) per year, including collection, treatment, and disposal/reuse. As mentioned previously, the total population equivalent is 10,000 PE. Schematic diagrams depicting comprehensive system boundaries for each scenario are shown in Section 2.1.1. The system boundaries encompass the collection and management of wastewater (foreground processes), as well as the production and transportation of chemicals, electricity, heat, and infrastructure (background processes). Additionally, all scenarios factor in avoided processes pertaining to fertilizer, biogas, and reclaimed water production. The substitution of these resources will influence the fertilizer and biogas market in terms of production, supply, and price. For example, the demand for electricity in the studied region affects the production mix, with the same applying to the mineral fertilizer market. In consideration of these "foreseeable" impacts on energy and mineral fertilizer systems, CLCAs with marginal data are deemed most suitable.

2.2.2. Life cycle inventory (LCI)

The inventory, comprehensively detailed in the supplementary material (SM), spans a wide range of processes for each scenario. It includes a mass balance for each scenario, measuring inputs and outputs in each unit process. The inventory encompasses building collection (piping and porcelain), sewer infrastructure (piping, excavation, and backfilling), treatment facility operation (chemical and energy use), and facility construction. Furthermore, it models other unit processes such as biogas upgrading, sludge treatment, and fertilizer recovery, all of which are documented in the SM, along with the Ecoinvent processes used.

2.2.2. Life cycle impact assessment (LCIA)

We used the ReCiPe® 2016 method (Midpoint, World – Hierarchistic version) and Simapro® for modeling. We altered the impact categories and selected the five that were most significant to the assessment: Global warming potential (GWP) in kg CO2-eq, Stratospheric ozone depletion (SOD) in kgCFC11- eq, Terrestrial acidification (TAD) in kg SO2-eq, Freshwater eutrophication (FEP) in kg P-eq, and Marine eutrophication (MEP) in kg N-eq.

2.2.3. Sensitivity Analysis

Using sensitivity analysis in LCA studies allows us to determine the robustness of the results and their sensitivity to uncertainty. A common method used in LCAs is Monte Carlo, supported by software like Simapro®. However, in our case, the Monte Carlo method would not work properly due to the use of consequential system models. Hence, we carried out a sensitivity analysis in the form of scenarios on some uncertain but critical factors affecting the study's outcome. Our first scenario examined the NH_3 emissions from the urine recycling system. Initially, in line with the literature (Gao et al., 2024) (in preparation), it was assumed that NH_3 losses would not occur during concentration, and, hence, N, P, and K could be effectively concentrated up to 99 %. For the purpose of this sensitivity analysis, it was assumed that 5 % of the nitrogen may be lost as NH3 emissions during concentration. The second sensitivity scenario explored using acid agents other than citric acid for urea stabilization. (Simha et al., 2023) reported that the following acids: 1.36 g H2SO4 $L-1$, 2.86 g H3PO4 $L-1$, 2.53 g C2H2O4·2H2O $L-1$, and 5.9 g C6H8O7 $L-1$ were found to be effective for urine stabilization. Thus, this scenario will compare these alternatives in terms of their environmental performance and impact on the urine recycling system's GWP. Thirdly, we consider the use of electricity by the urine recycling system in its operation, and particularly the energy efficiency of the urine concentrator. The concentrator was assumed to recover 70 % of its energy demand (600 Wh/L) (Simha, 2021). In comparison, the sensitivity analysis considered a scenario in which no energy recovery was performed, and the system used 600 Wh per liter of urine. The fourth sensitivity scenario concerns the percentage of greywater recovered and utilized for irrigation purposes in the third and fourth systems. In line with the literature for similar studies (Lima et al., 2023), we assumed a recovery rate of 80 %, which may appear high for irrigation needs in typical urban areas, especially since the investment in storage systems is outside the scope of our study. Therefore, we proposed a sensitivity analysis that assumes a more conservative recovery rate of 40 %, with the remaining 60 % being discharged into the ocean. Finally, we considered different sources of electricity. The original scenarios accounted for the Swedish electricity mix. However, in this sensitivity scenario, we examined whether switching to the European energy mix would affect environmental impacts. These sensitivity scenarios test the robustness of the results drawn from the study and allow an understanding of how changes in these key parameters could have an impact on the overall environmental assessment.

3. Results and discussion

3.1. The comparative life cycle environmental impacts – RQ1

The characterized net results of the LCA are presented in Table 1. Upon initial examination, it is apparent that the fourth scenario, which incorporates a urine recycling system as well as a blackwater system, represents the best-performing sanitation system regarding GWP, ozone depletion, acidification and eutrophication in our study. In addition, it is evident that the inclusion of the urine recycling system in the second scenario significantly improved the WWTP's performance regarding these factors, resulting in a 20 % reduction in global warming potential, a 65 % reduction in ozone depletion, a 45 % reduction in acidification, and a 55 % reduction in marine eutrophication. It is crucial to clarify that the focus of this paper is not on predicting how WWTP managers would handle a technological system incorporating local urine recycling. Such predictions are outside the scope of this paper. WWTP managers would likely focus on meeting current demands on discharges, which will become even more manageable with local urine recycling due to lower incoming nitrogen and, thus, lower aeration requirements and chemicals in WWTPs (Kleckers, 2023). However, it is equally conceivable that stricter discharge limits could be implemented in the future to counterbalance this effect. Hence, authorities would likely seek to regulate the impact on WWTPs stemming from such technological advancements. Therefore, this paper explicitly investigates "the potential effect" of local urine recycling without considering the "potential policy or regulatory changes" necessary for a system with local treatment of urine or blackwater.

Table 1 presents the net results; each system's savings (negative emissions) from the substituted resources have not been explicitly delineated as they are already accounted for in the net. For a more comprehensive visualization of these gains and each unit process's contribution, see Fig. 4. It is evident therein that the positive emissions for the fourth scenario (92.6 and 1.3 kg CO2-eq per PE/ y) can be attributed to the treatment operation and construction, respectively. However, the system also has negative emissions, reflective of gains derived from the substitution of resources. For instance, - 54.5, −15.0, and - 0.5 kg CO2-eq per PE/ y from the NPK fertilizer, irrigation, and sludge fertilizer, respectively.

GWP values observed in the baseline scenario align with those documented in the literature (Besson et al., 2021; Diaz-Elsayed et al., 2020; Spångberg et al., 2014; Thibodeau et al., 2014). It is necessary to

Table 1
Complete characterized life cycle assessment results using the ReCiPe® method (ReCiPe-LCA) for the conventional WWTP and source-separating sanitation systems. Highlights represent the best-performing results.

Impact category	Unit	S1: Conv. WWTP	S2: Urine +WWTP	S3: BW&GW	S4: Urine +BW&GW
Global warming	kg CO2-eq per PE/y	78	62	32	24
Stratospheric ozone depletion	kg CFC11 eq per PE/y	8.2E-04	2.9E-04	6.0E-05	4.7E-05
Terrestrial acidification	kg SO2 eq per PE/y	3.3E-01	1.8E-01	7.2E-02	-1.2E-01
Freshwater eutrophication	kg P eq per PE/y	8.8E-03	7.0E-03	-2.0E-03	-1.2E-02
Marine eutrophication	kg N eq per PE/y	5.0E-01	2.2E-01	2.7E-02	2.9E-02

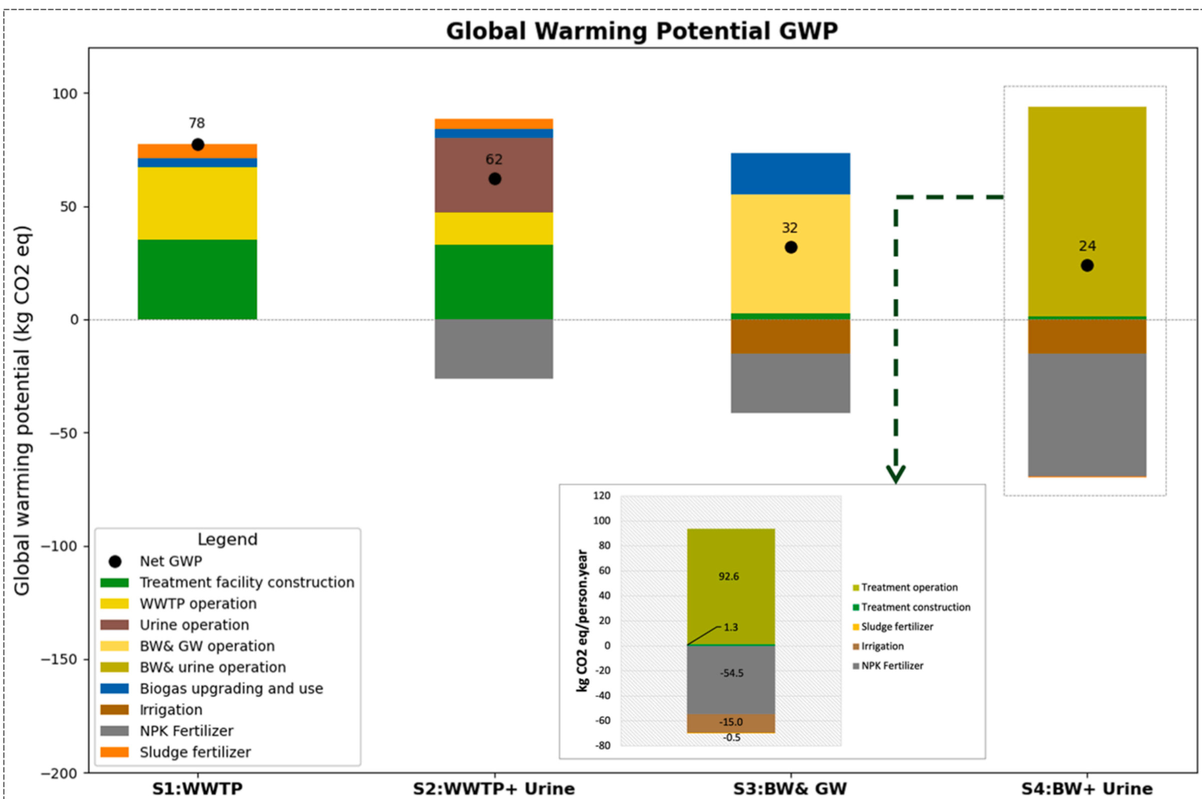

Fig. 4. The global warming potential GWP net results of the analyzed systems using the ReCiPe® method. The units are in kg CO2-eq per PE/ year. The fourth scenario has been broken down to show detailed results.

emphasize that discrepancies between LCAs may arise for several reasons. A crucial determinant is the nature of the data used, as discussed previously, where disparities may result from the utilization of marginal versus average datasets (Corominas et al., 2020). Additionally, the delineation of system boundaries within the LCA framework and the district typology exerts a significant influence on potential outcomes. Furthermore, the inclusion of recovered resources in the assessment process, the specific LCA methodology employed, the configurations of the analyzed systems—which can affect critical parameters such as N_2O emissions—and the energy sources utilized all play a significant role in shaping the assessment results (Diaz-Elsayed et al., 2019; Lehtoranta et al., 2022b). The assessment of other source-separating sanitation systems is also subject to similar considerations (Corominas et al., 2013).

To compare the four scenarios concerning the comprehensive array of other impact categories outlined in the table, the corresponding values have been plotted and illustrated in *Fig. A.5*. As previously indicated, the fourth scenario demonstrates the most modest impact across all categories assessed. Notably, this scenario manifests total negative values for two impact categories: acidification and freshwater eutrophication. Negative impacts are largely due to the utilization of NPK fertilizer, biogas, and reclaimed water.

3.2. Environmental hotspot identification and mitigation recommendations

In the initial scenario, the GWP is estimated at 78 kg CO2-equivalent per PE/ y. For a more detailed understanding of these emissions, Fig. 5 illustrates the unit processes that were modeled. It is evident from the figure that the majority of GWP is generated by the operation and construction of the WWTP. A more detailed analysis is provided within the same figure by depicting the operation unit process. In addition to electricity consumption, nitrous oxide from biological nitrogen removal in the activated sludge system and methane emissions from the anaerobic digester during biogas production also contribute significantly, accounting for 30.3 and 15.2 kg of CO2-equivalents per PE/ y, respectively. Furthermore, it is noteworthy that the integration of recovered heat, intended to replace conventional district heating, has shown positive results. This substitution has resulted in a reduction of −17.7 kg CO2-equivalent per PE/ y. The ozone depletion potential of the WWTP was calculated at 8.2E-04 kg CFC11 eq per PE/ y, the highest in comparison to the other scenarios (see Fig. A.5). A major contributor to ozone depletion is sludge management and nitrogen oxide emissions at the treatment plant. The management of sludge also contributes significantly to acidification due to emissions of ammonia (NH3), nitrous oxide (N2O), and methane (CH4). The eutrophication category is further divided into freshwater and marine, which reflect nutrient emissions (P and N, respectively) from the WWTP into the water. The results showed 0.499 kg N per PE/ y and 0.024 kg P per PE/ y, equivalent to 9.49 mg N / L and 0.45 mg P / L.

For the second scenario, integrating urine recycling with the WWTP resulted in 62 kg CO2-eq per PE/ y, which is a 20 % reduction of the WWTP GWP. To facilitate a comprehensive understanding of the scenarios, Fig. 6 illustrates the distinct stages that contribute to the GWP. It is evident from the figure that the introduction of urine recycling has significantly reduced the GWP of the WWTP operation from 32.3 (in the baseline scenario) to 14.3 kg CO2-eq per PE/ y. This is attributed mainly to a reduction in electricity required to treat the influent with lower nitrogen and phosphorus loads, consequently leading to a reduction in nitrous oxide emissions and methane emissions, similar to what was reported in (Besson et al., 2021). In addition, urine recycling led to a reduction in all other impact categories compared to the reference scenario, for example, there was a 55 % reduction in eutrophication potential caused by the decrease in nutrient discharge (N & P) into water bodies, especially the nitrate (NO_3-N) concentration, similar to what was reported in (Jimenez, 2015) . These findings align with the literature (Hilton et al., 2021), reporting that urine diversion and concentration could achieve a 29–47 % reduction in GWP and 25–64 % in eutrophication over conventional WWTP. Furthermore, there was a 65 % and 45 % reduction in ozone depletion and acidification potential, respectively (see Fig. A.5).

Moreover, the urine recycling system produces NPK fertilizer, which is assumed to replace mineral fertilizer. This substitution leads to a - 26.3 kg CO2-eq per PE/ y reduction in the scenario's GWP (Fig. 6). On the other hand, it is necessary to acknowledge that operating the urine treatment system contributes significantly to the GWP, illustrating the inherent trade-offs associated with many sanitation systems. Even though the urine recycling system brings gains, such as negative emissions via the replacement of mineral fertilizer, the operation of the urine recycling system in terms of energy demand and chemical use contributes to greenhouse gas emissions. A further investigation into the sources of GWP associated with urine recycling reveals that the urine concentrator and the stabilization tank constitute the primary contributors, contributing 16.22 and 15.48 kg CO2-eq per PE/ y, respectively. Among the main contributors is the use of citric acid as a stabilizing agent in the stabilization tank, which requires energy for the microbial fermentation and purification processes. Additionally, electricity consumption is a significant factor that affects urine concentrator performance.

For the third scenario, the black and greywater system, the total GWP was estimated to be 32 kg CO2-eq per PE/ y, a 60 % and 48 % reduction compared to the baseline scenario WWTP and the second scenario, respectively. Although these findings, i.e., a reduction in percentage from the baseline align with the literature (Kjerstadius et al., 2017; Lima et al., 2023), although the exact GWP values differed. This can be attributed to the type of system models used, the system boundaries, and the person equivalent. For a better understanding of the GWP, the different unit processes are illustrated in Fig. 7. The figure shows that the major contributors to the GWP are operation and biogas upgrading. The NPK and recovered water have negative GWP as gains (−15 and −26.3 kg CO2-eq per PE/ y) attributed to their mineral fertilizer and irrigation substitution. The operation unit process has been broken down to look at its inputs to better understand where the GWP comes from. The figure shows that ammonia stripping and struvite precipitation contribute to much of the operation GWP of 28.6 and 9.29 kg CO2-eq per PE/ y, respectively. This is attributed to the chemicals used in both processes (e.g., Sulfuric acid, Sodium hydroxide, and Magnesium chloride), which aligns with what is reported in the literature (Lima et al., 2023). Regarding other impact categories, this scenario outperforms the first two scenarios in all categories, and the system received gains, including negative emissions in acidification, ozone depletion, and eutrophication attributed to the utilization of NPK and sludge fertilizer, reclaimed water, and biogas use (see Fig. A.5).

The fourth scenario is a hybrid system that combines the urine recycling system with the black and greywater system. The total GWP has been reduced to 24 kg CO2-eq per PE/ y, which is attributed to the extra NPK recovered from the urine. This scenario achieves an almost 70 % reduction in GWP compared to the baseline scenario and a 22 % reduction compared to the BW scenario. As shown in Fig. 8, the negative GWP from NPK has increased from 26.3 for the BW without urine recycling to 54.5 kg CO2-eq per PE/ y with urine recycling. The treatment operation in this scenario contributes 92.6 kg CO2-eq per PE/ y. This includes the 33.2 kg CO2-eq per PE/ y from the urine recycling system and 18.03 kg CO2-eq per PE/ y from the biogas upgrading; thus, the BW operation is 41.4 kg CO2-eq per PE/ y, which is lower than in the

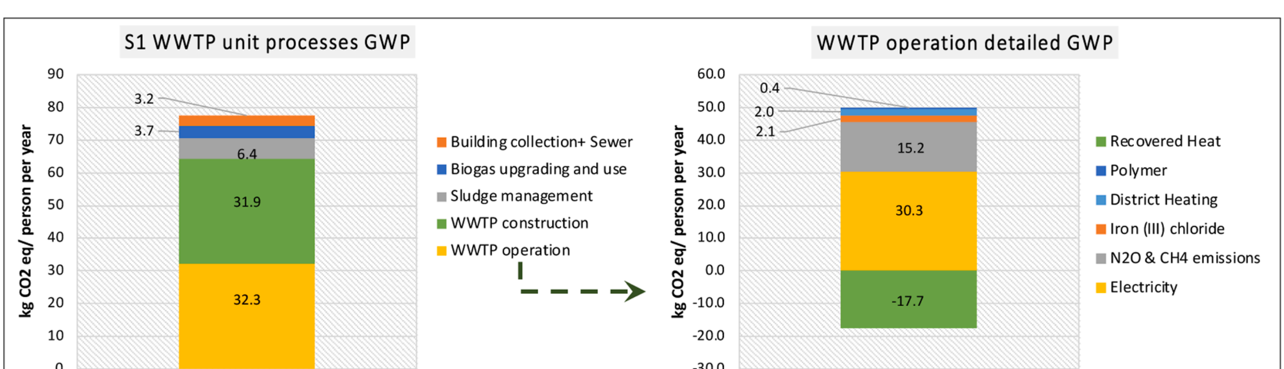

Fig. 5. Scenario 1, WWTP unit processes global warming results and the detailed WWTP operation unit process results.

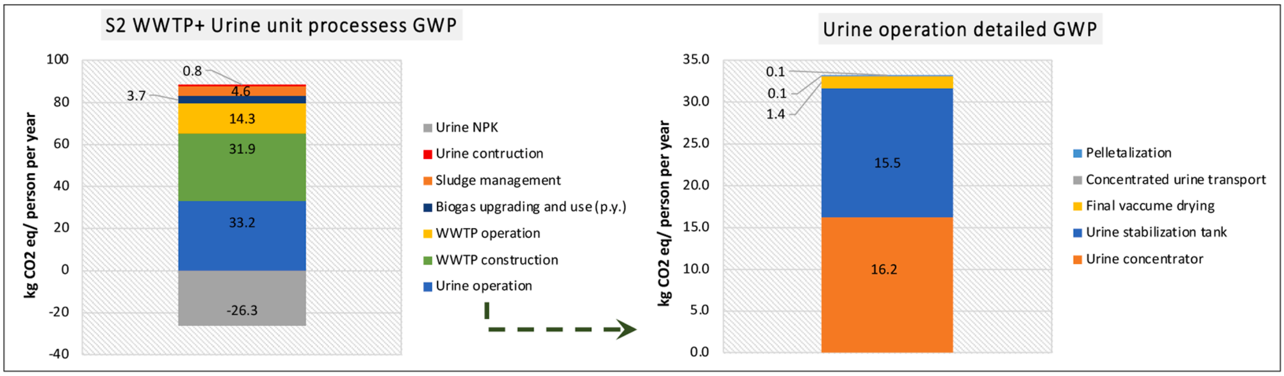

Fig. 6. Scenario 2. Urine and WWTP unit processes global warming results and the detailed Urine operation unit process results.

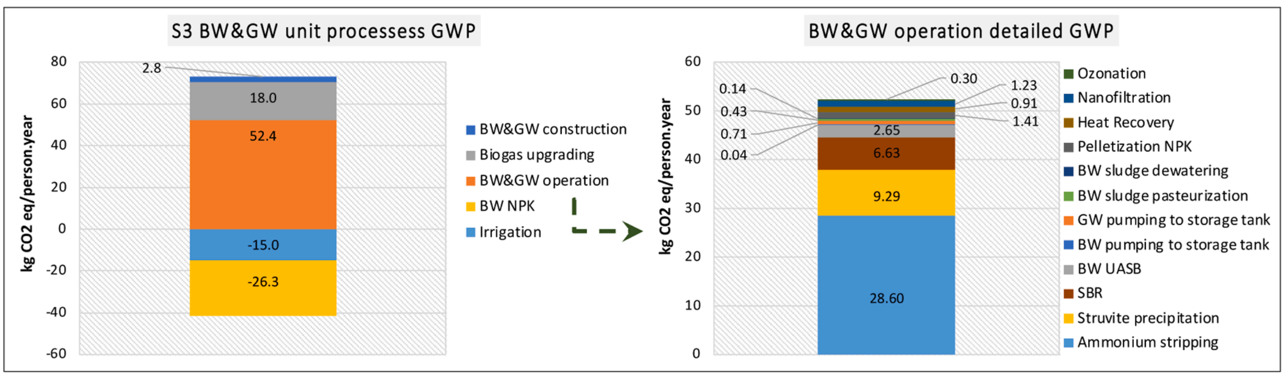

Fig. 7. Scenario 3, BW & GW unit processes global warming results and the detailed results of the operation unit process.

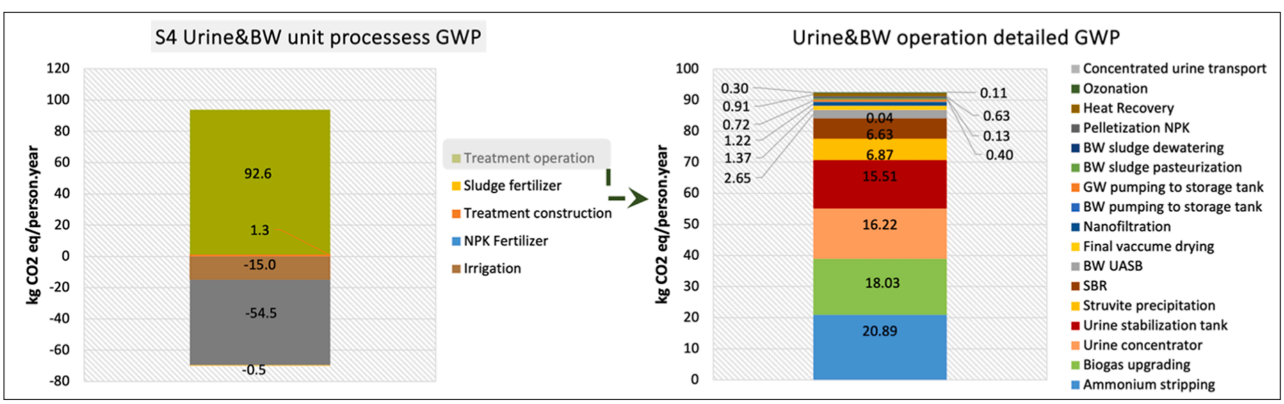

Fig. 8. Scenario 4, Urine and BW unit processes global warming results and the detailed results of the operation unit process.

third scenario without urine recycling. This is because urine recycling decreases the impact of the ammonia stripping and struvite precipitation processes, which had the highest share of the GWP in the third scenario. To better understand the treatment operation, we can see that ammonia stripping GWP is 20.89 kg CO2-eq per PE/ y compared to 28.6 kg CO2-eq per PE/ y in the third scenario. Regarding other impact categories, this scenario outperforms all scenarios in all categories, and the system received gains, including negative emissions in acidification, ozone depletion, and eutrophication attributed to the extra utilization of NPK and sludge fertilizer, reclaimed water, and biogas use (see Fig. A.5).

3.3. Sensitivity analysis

The first sensitivity scenario assumed 5 % NH_3 emissions at the urine concentrator, which is in opposition to the initial assumption of no ammonia loss. The changes exclusively affected the second and fourth scenarios, which incorporated urine recycling, while the first and third scenarios remained unchanged. The urine recycling system includes the following unit processes: urine stabilizer, concentrator, transport of concentrated urine, vacuum drying, and pelletization. The results revealed a slight (4 % and 8 %) increase in the GWP of the second and fourth scenarios. Additionally, there was a significant increase in their acidification potential by over 200 % and 300 % from 0.18 to 0.6 and −0.12 to 0.28 kg SO2 eq per PE/ y in the second and fourth scenarios, respectively. To compare these values with other scenarios, see Fig. A.5. SM contains further information regarding the impacts on other categories.

The second sensitivity analysis evaluated the environmental performance of the urine recycling systems using four different acid agents instead of citric acid. Sulfuric acid 1.36 g H2SO4 per liter of urine had the best environmental performance. Results showed that the whole GWP of the urine recycling system could be reduced by 47 % from 33.2 to 17.6 kg CO2-eq per PE/ y. The urine stabilization unit process had a 15.47 kg CO2-eq per PE/ y GWP when 10 g of citric acid was used. When sulfuric acid was used instead, the GWP was reduced by 101 % (negative savings) to −0.14 kg CO2-eq per PE/ y (see Fig. 9 for a detailed illustration). This is because sulfuric acid can be produced as a by-product in various industrial processes (e.g., copper smelting and desulfurization of crude oil), a practical and sustainable approach that improves the overall efficiency and sustainability of industrial operations. Thus, from a consequential perspective, the marginal emission factor for sulfuric acid is negative. However, there are challenges associated with the use of sulfuric acid that fall outside the scope of this LCA. Since sulfuric acid is a byproduct of fossil fuel production, transitioning to a fossil-free environment could lead to concerns about the availability of sufficient H2SO4, especially since current known minable resources are projected to last <30 years (Maslin et al., 2022).

Increasing the electricity demand in the urine recycling system to 600 Wh per liter of urine in the third sensitivity scenario resulted in a marked increase of almost 50 % in the GWP of the whole system, mainly coming from the urine concentrator unit process, which saw GWP increasing by 66 %, from 16.22 to 48.67 kg CO2-eq per PE/y. The latter sensitivity analysis made the scenarios incorporating urine (i.e., the second and fourth) look worse compared to the reference scenario and the BW.

For the fourth sensitivity analysis, we focused on the percentage of greywater GW recovered and utilized for irrigation. We used a more conservative recovery rate of 40 %, with the remaining 60 % being discharged into the ocean instead of the 80 % recovery in the initial scenario. The changes exclusively affected the third and fourth scenarios, which incorporated GW recycling, while the first and second scenarios remained unchanged. The one-unit process that was affected the most in both systems is the irrigation unit. Initially, both systems saved 15 kg CO2-eq per PE/y due to the recovered GW used for irrigation. However, when the recovery rate decreased to 40 %, the savings from irrigation also dropped to 7.5 kg CO2-eq per PE/y. Additionally, there was a slight change in the ozonation unit process due to the increased flow of GW out of the nanofiltration to the ocean, resulting in approximately 0.5 kg CO2-eq per PE/y. These two changes in the systems led to an increase in their global warming potential (GWP) to 40 and 32 kg CO2-eq per PE/y, as illustrated in Fig. A.6. Nevertheless, the two systems still outperformed the reference and second scenarios.

In the final sensitivity analysis, we examined the consequences of switching from the Swedish to the European energy mix. While all impact categories demonstrated an increase, the observed increase of approximately 10 % was less pronounced than anticipated. This deviation can be attributed to the utilization of marginal data in the consequential model (Wernet et al., 2016). When utilizing the marginal data in the consequential model, the model does not simply average out all EU power source mixes, such as coal, gas, nuclear, and renewables (Regett et al., 2018). Instead, the focus is on what power sources would actually increase production to meet the anticipated increase in demand or whether the increase would be met by imported electricity (Aliahmad et al., 2020; Vélez-Henao et al., 2019). The method used by Ecoinvent to develop marginal electricity data is to take a long-term forecast or scenario for future electricity production and define/assume the marginal electricity mix to be a mix of technologies, where the electricity is projected to increase from now until the future scenario (Ekvall, 2020; Regett et al., 2018). Supposing the trends identified for the EU marginal future electricity show a predominance of cleaner technologies (like wind or solar), the change in GWP might not be as high since these cleaner sources have lower CO2 emissions than coal (Naumann et al., 2024; Schmidt J H et al., 2011). However, this method has the drawback of ignoring declining trends; instead, it only accounts for growing ones. Based on the Ecoinvent v3 database, the average emission factor for the European electricity mix is 0,39 kg CO2-eq; however, the marginal emission factor is 0,21 kg CO2-eq, which implies that the modeled trend for the EU future electricity production and expansion is predominated by clean technologies. In conclusion, these results for the sensitivity analysis in Fig. A.6 indicate that the framework of assessment of this LCA and the data modeled in the inventories are robust and that the sensitive parameters considered are of high significance in terms of their contribution to the different impact categories.

3.4. Comparative analysis and practical insights

The conventional WWTP modeled as the reference scenario showed the highest environmental impact across all assessed impact categories. This underscores the necessity for innovations that contribute to a reduction in the environmental impacts of conventional systems, especially at the biological nitrogen removal stage. Scenario 2, which incorporates a urine recycling system, demonstrated improvements over the conventional WWTP and can thus be a coherent pathway toward sustainable improvement. In this scenario, the nitrogen and phosphorus load on the treatment plant decreased, correspondingly lowering the energy demand for biological nitrogen removal and the dosage of chemicals required for precipitating phosphate. Furthermore, fertilizer recovery from urine recycling reduced GWP and eutrophication impacts. Scenario 3, the BW&GW system, demonstrated further improvements compared to the conventional WWTP and urine recycling system. Nutrient recovery, biogas production, and reclaimed clean water significantly reduced its GWP and attained excellent results across all assessed impact categories. Additionally, treating greywater locally in this scenario reduces the load (i.e., the volume of wastewater) to the centralized WWTP, thus enhancing the treatment plant's efficiency and capacity, particularly during peak periods (Awasthi et al., 2024). In the

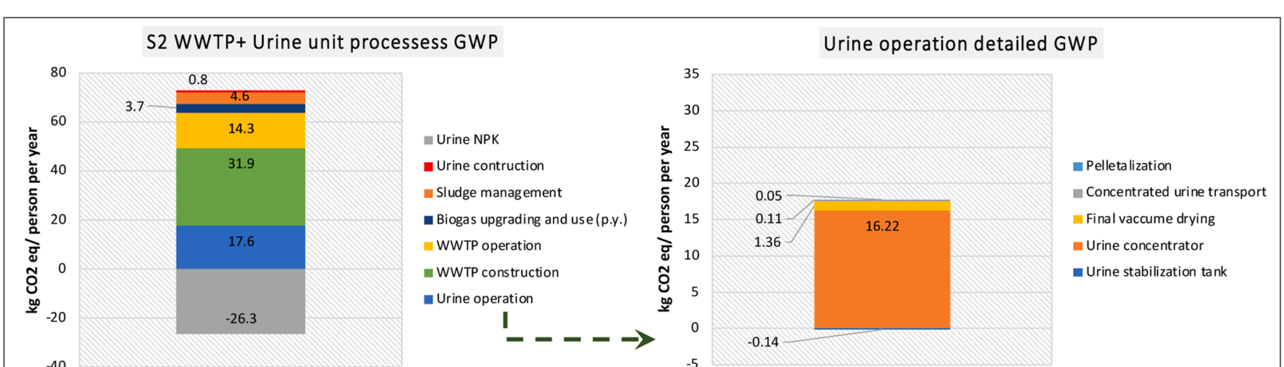

Fig. 9. Detailed analysis of the impact of using sulfuric acid instead of citric acid. The GWP of urine operation dropped from 33.23 in the second scenario to 17.61 kg CO2-eqper PE/ y. The primary reduction is in the stabilization tank (100.88% reduction), from 15.47 kg to −0,137 kg CO2-eqper PE/ y.

fourth scenario, the hybrid system, particularly when combined with 70 % heat energy recovery at the urine concentrator, showcased the best environmental performance among all other scenarios across all assessed impact categories. Integrating the urine recycling system with the BW&GW system offers a more holistic and completely decentralized approach that maximizes resource recovery, reduces the GWP of the energy-intensive ammonia stripping process, and enhances all other assessed impact categories.

In real-world applications, various factors, including infrastructure availability, resource recovery targets, social acceptance, and the environmental conditions of the local water recipients, will guide the choice of sanitation systems. Decision-making considerations are recommended to focus on the accruable long-term benefits from reduced environmental impacts, resource recovery, and energy generation when choosing appropriate sanitation systems. This paper is based on a sound framework for assessing the environmental profiles of different sanitation scenarios and, hence, forms a key instrument in guiding sustainable wastewater management practices. For already-built neighborhoods connected to sewer networks and centralized treatment plants that will undergo renovation, we recommend integrating urine recycling into the new units. This is a crucial step towards sustainability, as it promises considerable improvements to the treatment plant, including a reduction in its environmental impacts, increased capacity, and local production of bio-based fertilizers that contribute to food security and nutrient resilience. For unbuilt neighborhoods in the planning stage, we recommend a completely decentralized source-separating system like the BW&GW system, which offers a promising reduction across all investigated impact categories. To further optimize the environmental profile and sustainability of the neighborhoods, we recommend the hybrid scenario, which integrates urine recycling with 70 % energy recovery at the urine concentrator into the BW&GW system. Hence, the priority should not be deciding between urine recycling and the BW&GW system but instead integrating both for a more comprehensive decentralized source-separating sanitation solution.

4. Conclusion

This study conducted a comprehensive consequential life cycle assessment (LCA) utilizing marginal data and system expansion/substitution to compare the environmental performance of various source-separating sanitation systems to that of a centralized wastewater treatment plant (WWTP). The centralized WWTP served as the reference scenario. The second scenario included urine recycling integrated into the reference scenario. The third scenario examined the implementation of a black and greywater (BW & GW) system. Finally, the assessment featured a hybrid scenario that combined urine recycling with the BW & GW system.

Results indicated that the Global warming potential GWP of the four scenarios were estimated to be 78, 62, 32, and 24 kg CO2-eq per PE/ y, respectively. The findings suggest that integrating a urine recycling system into the WWTP could potentially reduce GWP by 20 %. This reduction is primarily attributed to the gains and savings from the recovered NPK fertilizer, which would effectively replace mineral fertilizer. The black and greywater system (BW & GW) in the third scenario achieved a significant 60 % reduction over the reference scenario and 48 % over the second. This reduction is largely attributed to the savings and gains from recovering NPK fertilizer, biogas, and clean water, which serve as alternatives to mineral fertilizer, diesel, and irrigation water. For the hybrid system in the fourth scenario, integrating the urine recycling system into the BW system reduced the GWP by almost 70 % compared to the baseline scenario and 22 % to the third scenario. The reduction in the BW system is primarily attributed to the mitigation of the GWP associated with ammonia stripping, which is due to its high energy and chemical demands. Hence, utilizing urine recycling to manage nitrogen flows instead of ammonia stripping leads to a notable decrease in the GWP of the BW system. The urine recycling system also contributed to additional gains through NPK fertilizer recovery. The potential impact of using different chemicals for urine stabilization was also examined, with results suggesting that switching from citric acid to sulfuric acid could potentially reduce the stabilization unit process GWP by 101 %, bringing the impact down from 15.47 to −0.14 kg CO2-eq per PE/ y.

It's essential to remark that the performance of source-separating systems is largely attributed to the resources these systems recover, which translate into savings from their total GWP and give these systems an edge to outperform conventional systems. The recovery of resources is subject to assumptions and requires a thorough examination of their uncertainty and sensitivity, particularly concerning the considerable savings, such as those achieved through the recovery of fertilizer, biogas, and water, which significantly impact the overall outcomes. For instance, the sensitivity analysis revealed that lowering the recovery rate of greywater to 40 % instead of 80 % reduced the gains in the third and fourth scenarios by 7.5 kg CO2-eq per PE/ y. Although the two systems still outperformed the reference scenario, their total GWP increased to 40 and 32 kg CO2-eq per PE/y.

In conclusion, the BW & GW system in the third scenario emerged as a great environmental choice compared to the centralized WWTP. However, the additional benefits of the urine recycling system in both the BW and WWTP make it an essential component in choosing sustainable sanitation solutions. Ultimately, the findings suggest that the fully decentralized source-separating system incorporating urine, BW, and GW recycling, as demonstrated in the fourth scenario, is the most favorable environmental profile. This implies that when it comes to source separation, the critical factor is not simply a selection between urine and blackwater systems. Instead, it suggests that a hybrid or integrated source-separating system offers the most promising environmental performance and sustainability benefits, particularly when combined with 70 % energy recovery at the urine concentrator.

CRediT authorship contribution statement

Abdulhamid Aliahmad: Writing – review & editing, Writing – original draft, Visualization, Validation, Software, Methodology, Formal analysis, Data curation, Conceptualization. **Priscila de Morais Lima:** Writing – review & editing, Supervision, Resources, Data curation, Conceptualization. **Hamse Kjerstadius:** Writing – review & editing, Supervision, Resources, Data curation, Conceptualization. **Prithvi Simha:** Writing – review & editing, Supervision, Data curation, Conceptualization. **Björn Vinnerås:** Writing – review & editing, Supervision, Data curation, Conceptualization. **Jennifer McConville:** Writing – review & editing, Validation, Supervision, Funding acquisition, Data curation, Conceptualization.

Declaration of competing interest

The authors declare the following financial interests/personal relationships which may be considered as potential competing interests:

Jennifer McConville reports financial support was provided by Swedish Research Council Formas. If there are other authors, they declare that they have no known competing financial interests or personal relationships that could have appeared to influence the work reported in this paper.

Supplementary materials

Supplementary material associated with this article can be found, in the online version, at doi:10.1016/j.watres.2024.122741.

Appendices

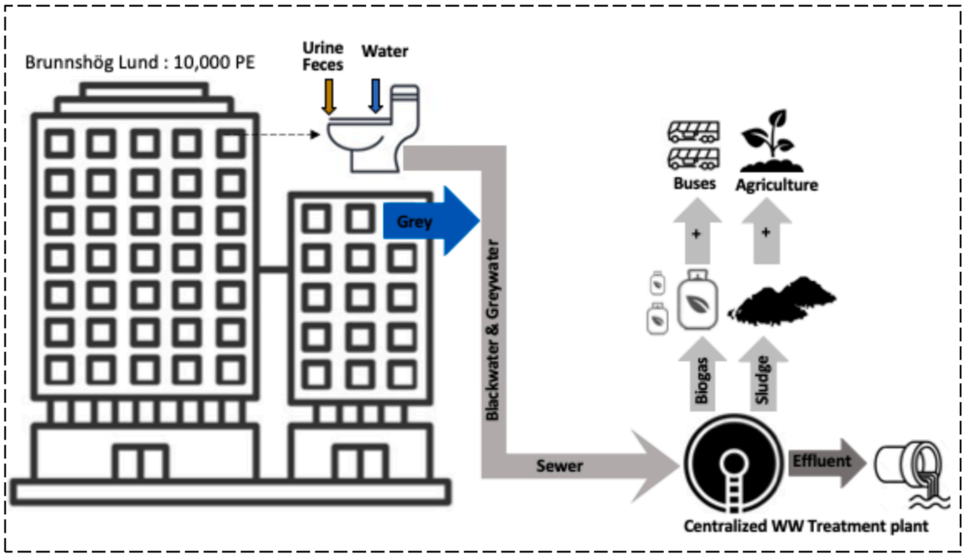

Fig. A.1. The layout of the first scenario, conventional WWTP. All wastewater fractions are mixed and transported in one pipe to the plant. The treatment plant treats influent and produces biogas and sludge that can be used in buses and agriculture. Effluent is discharged into a local water body.

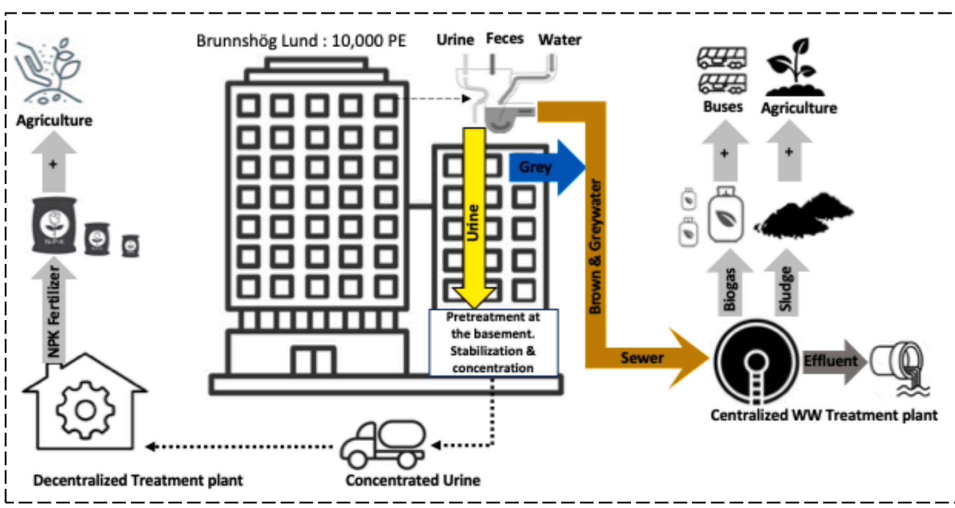

Fig. A.2. The layout of the second scenario, urine recycling + conventional WWTP. Urine is collected separately using a diversion toilet, and then the rest of the wastewater is collected, mixed, and transported in one pipe to the plant. The urine is pretreated in the basement and later treated to produce NPK fertilizer.

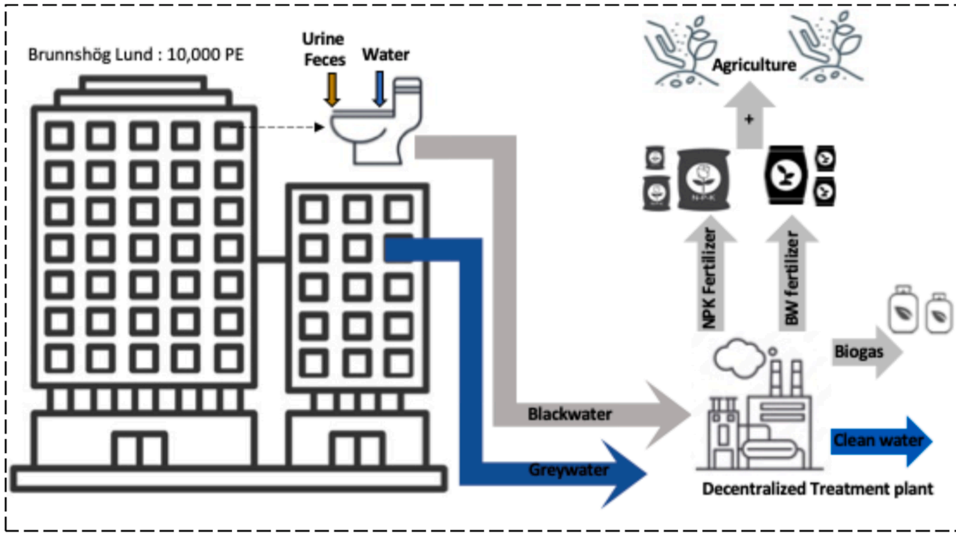

Fig. A.3. The layout of the third scenario, black and greywater. Blackwater and greywater are collected separately using two pipes. Each fraction is treated separately in the on-site treatment plant. NPK fertilizer, biogas, and clean water are produced.

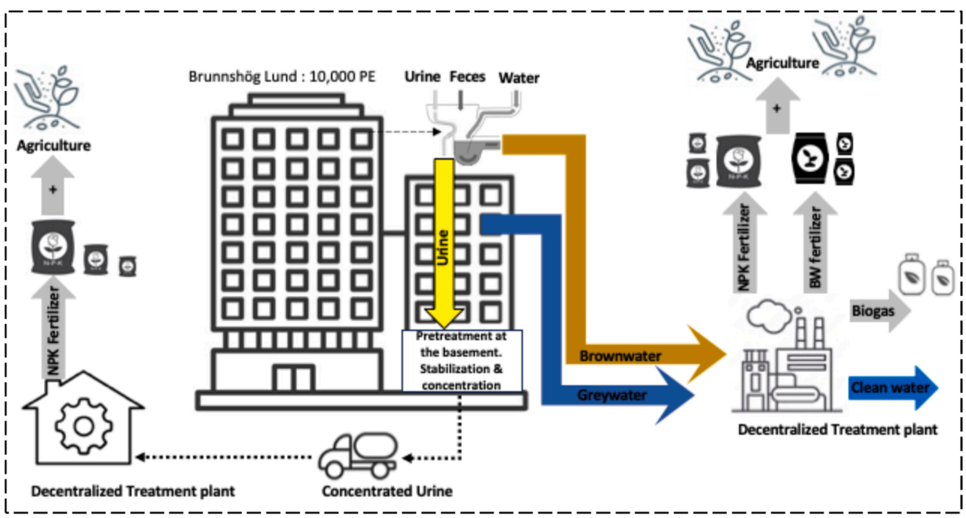

Fig. A.4. The layout of the fourth scenario, urine recycling + black and greywater. Urine (75 %) is collected separately using a diversion toilet. The brown water and the 25 % left of urine are collected separately in a second pipe; the greywater is also collected separately in a third pipe. Each fraction is treated separately, and NPK, biogas, and clean water are produced.

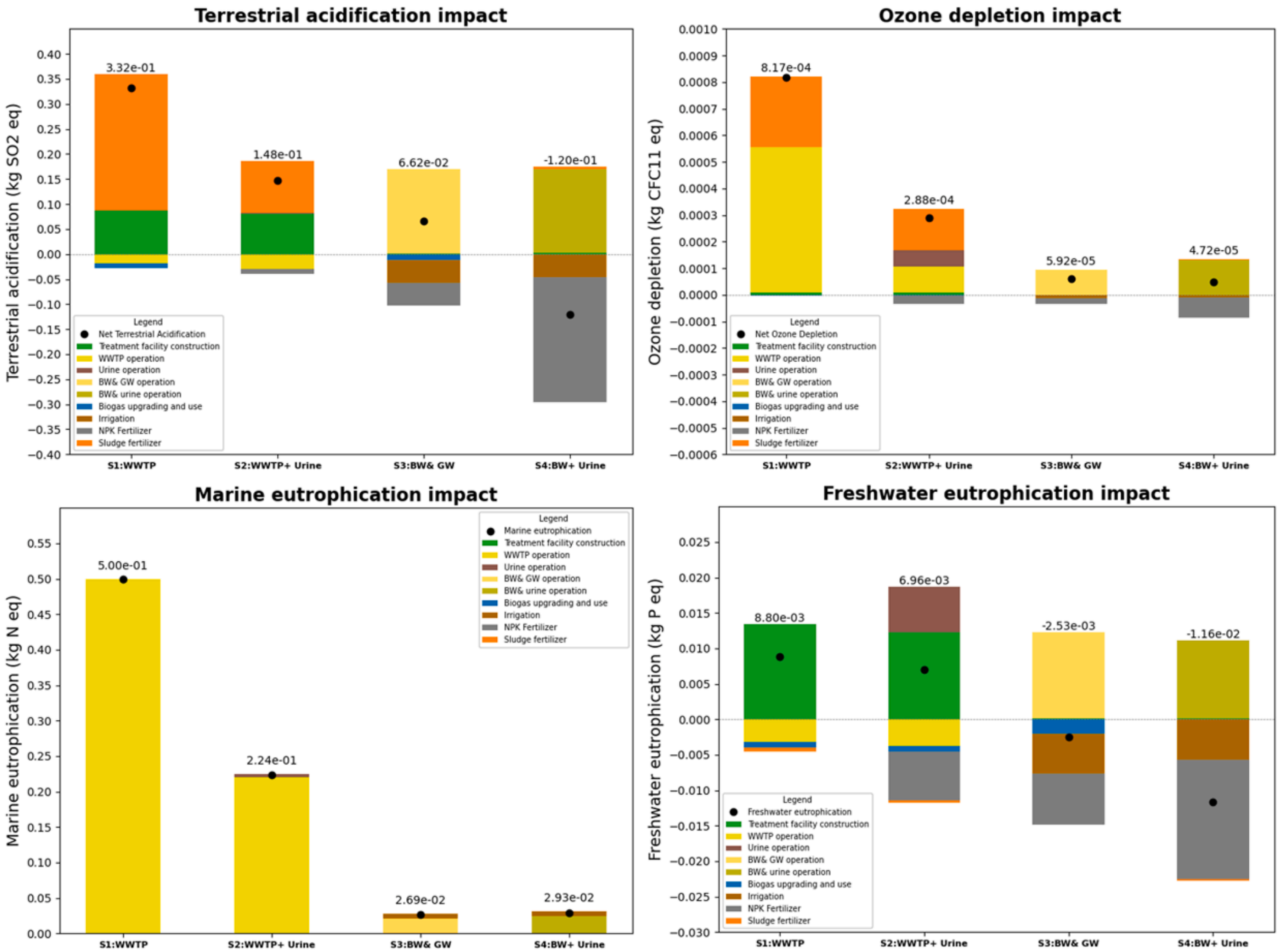

Fig. A.5. The net results of the analyzed systems using the ReCiPe® method.

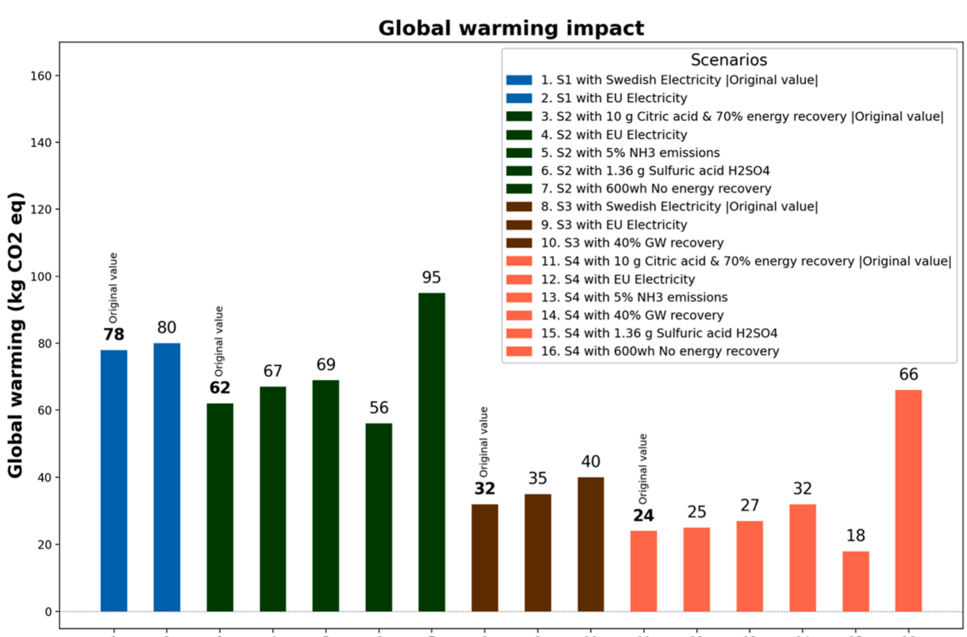

Fig. A.6. The GWP of the different sensitivity scenarios for the analyzed systems.

Data availability

All data can be found in the supplementary material submitted along with the research paper.

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